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The Effect of Wastewater Phosphorus Removal on Shagawa Lake, Minnesota: Phosphorus Supplies, Lake Phosphorus and Chlorophyll A

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16. ABSTRACT

In early 1973, the phosphorus supply to Shagawa Lake, Minnesota, was reduced by about 80% when a tertiary wastewater treatment plant began operating. Significant reductions in total and soluble reactive phosphorus concentrations have occurred in the lake since that time. By 1976 the average (volume weighted over the entire lake total and soluble reactive phosphorus concentrations had declined from about 51 and 21 $\mu g\ l^{-1}$ to about 30 and 4.5 $\mu g\ l^{-1}$, respectively, corresponding to 40 and 80 reductions. During 1975 and 1976, chlorophyll a (averaged over the top 5 m) had decreased to less than 50% of the pretreatment level during May-June but during July August little change had occurred.

A phosphorus residence time model projected equilibrium total phosphorus concentrations of about 12 $\mu g\ l^{-1}$ within 1.5 years. The fact that this level was not reached is attributed to a feedback of phosphorus from the sediments, primarily during summer. This phenomenon has been incorporated into a modified total phosphor mass balance model which projects the phosphorus pattern through 1976 quite accurately. The close fit of the modified mass balance model implies that the phosphoru supply from the sediments has not diminished since treatment began, and that further recovery of the lake will depend upon how long this feedback of phosphorus from the sediments continues.

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THE EFFECT OF WASTEWATER PHOSPHORUS REMOVAL ON SHAGAWA LAKE, MINNESOTA: PHOSPHORUS SUPPLIES, LAKE PHOSPHORUS AND CHLOROPHYLL *A*

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Abstract—In early 1973, the phosphorus supply to Shagawa Lake, Minnesota, was reduced by about 80% when a tertiary wastewater treatment plant began operating. Significant reductions in total and soluble reactive phosphorus concentrations have occurred in the lake since that time. By 1976 the average (volume weighted over the entire lake) total and soluble reactive phosphorus concentrations had declined from about 51 and 21 $\mu g\,l^{-1}$ to about 30 and 4.5 $\mu g\,l^{-1}$, respectively, corresponding to 40 and 80% reductions. During 1975 and 1976, chlorophyll *a* (averaged over the top 5 m) had decreased to less than 50% of the pretreatment level during May–June but during July–August little change had occurred.

A phosphorus residence time model projected equilibrium total phosphorus concentrations of about 12 $\mu g\,l^{-1}$ within 1.5 years. The fact that this level was not reached is attributed to a feedback of phosphorus from the sediments, primarily during summer. This phenomenon has been incorporated into a modified total phosphorus mass balance model which projects the phosphorus pattern through 1976 quite accurately. The close fit of the modified mass balance model implies that the phosphorus supply from the sediments has not diminished since treatment began, and that further recovery of the lake will depend upon how long this feedback of phosphorus from the sediments continues.

INTRODUCTION

Shagawa Lake has exhibited characteristics of cultural eutrophication for many years, in contrast to most other lakes in the forested region of northeastern Minnesota. Surface (average over the top 5 m) concentrations of total phosphorus (TP) and soluble reactive phosphorus (SRP) have exceeded 80 $\mu g\,l^{-1}$ and 30 $\mu g\,l^{-1}$, respectively, during summer (July–August). Springtime TP has been 50–60 $\mu g\,l^{-1}$ prior to inflow P reduction. Chlorophyll *a* concentrations reached 60 $\mu g\,l^{-1}$ during August and September and in isolated cases exceeded 100 $\mu g\,l^{-1}$. Anaerobic conditions developed in the deep waters during July and August as well as prior to ice-out in later winter (February–May). Blue–green algae bloomed throughout June–August forming noxious scums upon death and decomposition.

The development of these characteristics is attributed to a supply of wastewater nutrients (mainly P) from Ely, a nearby community of 5000 residents. Pilot plant and algal assay experiments indicated that wastewater with TP content reduced to 50 $\mu g\,l^{-1}$ or lower, when diluted with Burntside River water (the major influent to Shagawa Lake) in proportion to flow, would no longer support the growth of significant amounts of algae (Powers *et al.*, 1972). Approximately 80% of the external TP supply came from Ely's wastewater while only ~20% of the total nitrogen input was attributed to that source (Malueg *et al.*, 1975). Background details of the project and the lake's limnology can be found in: Bradbury (1978), Malueg *et al.* (1975), Megard & Smith (1974), Larsen *et al.* (1975), Larsen & Malueg (1976) and Schults *et al.* (1976).

In early 1973, the supply of TP to Shagawa Lake was reduced by ~80% when a tertiary wastewater treatment plant began operating (construction and operation were funded by the US Environmental Protection Agency [EPA] and the city of Ely). Since then, with minor exceptions, the treatment plant has maintained a TP effluent concentration of 50 $\mu g\,l^{-1}$ or lower. The treatment plant was designed primarily to remove wastewater TP, leaving the concentrations of other elements unaffected. The EPA's objective was to test the hypothesis that a major reduction in the supply of TP to a lake would alleviate the symptoms characteristic of eutrophy in that lake. Inevitably, the concentrations of other elements were somewhat altered by the treatment process, but none of these changes were as large as the reduction in TP supply (EPA, unpublished).

An earlier paper documented the changes, resulting from treatment, in TP supply, lake TP, SRP, and chlorophyll *a* through mid-1974 (Larsen *et al.*, 1975). This paper will extend those observations through

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1976 and describe in greater detail the changes in external supplies of TP and SRP.

METHODS

A form of the P residence time model was used to predict how TP in Shagawa Lake would be expected to change after the external TP supply was reduced (Megard, 1971; Sonzogni *et al.*, 1976; Vollenweider, 1969, 1975) and to evaluate the lake's response. The changes in TP in the lake can be described by:

$$\frac{d[TP]}{dt} = J/V - Q[TP]/V - S/V \qquad (1)$$

for a well mixed system having a net loss of TP to the sediments. [TP] is the lake TP concentration (mg m^{-3}); J, the rate of supply from external sources (mg $week^{-1}$); Q, the rate of water outflow (m^{-3} $week^{-1}$); V, the lake volume (m^3); and S, the net sedimentation rate (mg $week^{-1}$). The model assumes a constant lake volume and that water inflow and outflow are equal. The sedimentation loss term is often expressed as a fractional loss of lake TP: $S/V = \sigma_p[TP]$ where σ_p is the sedimentation coefficient (Vollenweider, 1969, 1975). The approach is best suited to describe long term changes. Annual values are often used for the variables in equation 1.

If J, S, and Q are assumed constant, the steady state solution of 1 can be written:

$$[TP] = J/Q(1 - R_p) \qquad (2)$$

where R_p, the TP retention coefficient, is the fraction of inflowing TP which is lost to the lake bottom (Dillon & Rigler, 1974a; Rigler, 1975; Vollenweider, 1975, 1976; Larsen & Mercier, 1976). J/Q is the average inflowing TP concentration.

Algal biomass (as chlorophyll *a*) produced in temperate lakes has been related to various measures of TP content in the lakes. Nicholls & Dillon (1978) have summarized some 16 relationships generated by a number of investigators. Generally the relationships correlate summer chlorophyll *a* (averaged over the period of thermal stratification) with springtime TP or annual TP, or correlate annual chlorophyll *a* with annual TP. Assuming equilibrium TP is similar to springtime TP (see Chapra & Tarapchak, 1976), Dillon & Rigler's (1974b) relationship between summer chlorophyll *a* and springtime TP was used to predict average summer chlorophyll *a* content from projected equilibrium TP content (equation 2).

TP dynamics in Shagawa Lake are dominated by a significant feedback of TP from the sediments twice each year: one small pulse develops during the spring and a larger pulse of greater duration occurs during July and August. Both pulses appear to be related to the generation of anaerobic conditions in the bottom waters of the lake (Larsen *et al.*, 1979), although the littoral could also be important, particularly during the summer (Lie, 1977). Equation 1 was modified to include this feedback because it has been a significant factor controlling the response of Shagawa Lake:

$$\frac{d[TP]}{dt} = J/V - Q[TP]/V - \sigma_p[TP] + J_{int}/V \qquad (3)$$

where J_{int} is an estimate of the rate of internal TP supply. J_{int}, σ_p, and the time interval over which TP was supplied internally were estimated from the pretreatment 1971–1972 data base. Initially, σ_p was assumed constant but it became clear that seasonal values (ice-free and ice-covered seasons) provided a better qualitative fit to the pretreatment data.

Specifically, J_{int} was assumed to have the following form:

$$J_{int} = \begin{cases} C_1,\ t_1 \le t \le t_2 \text{ (spring pulse)} \\ C_2,\ t_3 \le t \le t_4 \text{ (summer pulse)} \\ 0, \text{ all other times of the year,} \end{cases}$$

where C_1 and C_2 are constants (kg $week^{-1}$) and $t = 1, 2, \ldots 52$ weeks. σ_p was assumed to have two constant values. A winter rate, σ_p^w, applied to the periods when the lake was ice-covered, and a summer rate, σ_p^s, applied to the ice-free periods.

To estimate J_{int}, σ_p^w, and σ_p^s, computer simulations of equation 3 were performed and model trajectories were compared with data time series. For the data series, we partitioned the pretreatment interval from the onset of ice-cover in late 1971 until the onset of ice-cover in 1972 into an ice-covered and an ice-free period. The 1971–1972 ice-covered interval (23 weeks) was used to estimate σ_p^w, C_1, t_1 and t_2; the ice-free interval during 1972 (29 weeks) was used to estimate σ_p^s, C_2, t_3 and t_4. The rest of the terms in equation 3 come from measurements (Q, J, V). The remainder of the pretreatment period (the first 46 weeks of 1971 and the last six weeks of 1972) was used to corroborate the model.

The simulated trajectory was fit to the data by minimizing the sum of the squared deviations (SS) between the model and data. That is, SS is defined as

$$SS = \sum_k (TP_k^{model} - TP_k^{data})^2,$$

where the TP_k are weekly TP concentrations.

Initial guesses for the parameters J_{int}, σ_p^w, and σ_p^s were made. Then a computer algorithm employing the Davidon-Fletcher-Powell conjugate descent method (Pierre, 1969) iteratively adjusted the parameter values to decrease SS until values were found which minimized SS. Gradients of SS with respect to the parameters were calculated using the variational techniques of Hasdorff (1976). Details of the estimation procedures are given in Van Sickle & Larsen (1978).

Equation 3 was used to project the TP pattern which could be expected in the absence of treatment as well as to project the lake's response to treatment.

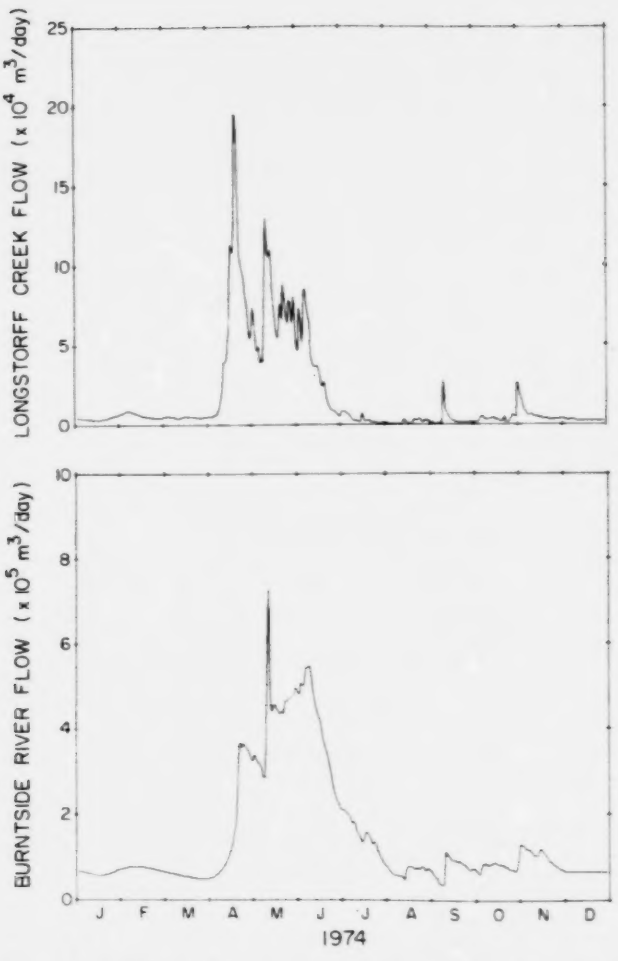

Fig. 1. Typical seasonal flow patterns for Burntside River and Longstorff Creek, influent tributaries to Shagawa Lake.

The methods by which the lake and tributaries were sampled for flow, TP and chlorophyll *a* (corrected for phaeophytin *a*) and the chemical methods by which these variables were measured have been described in Malueg *et al.* (1975), Larsen *et al.* (1975) and Larsen & Malueg (1976). Lake SRP might be underestimated because anaerobic samples might have become oxygenated during handling although precautions were taken to fill sample bottles completely and store them in the dark on ice until analysis within several hours of sampling. Oxygenation could cause precipitation of iron and phosphorus resulting in underestimates of actual SRP.

RESULTS

Influent phosphorus characteristics

Tributary inflow. Typical stream discharge patterns summarized for two tributaries, Burntside River and Longstorff Creek, demonstrate the seasonality of surface flow in the Shagawa Lake watershed (Fig. 1). Burntside River flow accounts for most water entering Shagawa Lake, typically 70–80% (Malueg *et al.*, 1975). Longstorff Creek (~ 10% of the inflow) is representative of the smaller tributaries. Most of the flow (all sources including rainfall and runoff) occurs during spring and early summer. Of the total yearly inflow during 1974–1976 approximately 60% entered the lake during the three-month period from mid-April to mid-July. Similar flow patterns occurred during pretreatment years. In 1973 the flow was relatively low during the spring and early summer runoff months (about 30% of the annual flow) with higher than average discharge during the fall, adding up to a normal annual flow.

Total annual flow to Shagawa Lake (Table 1) ranged nearly threefold over the eight-year interval of record (1969–1976). A low of 42.8×10^6 m^3

Table 1. Changes in phosphorus inflow and outflow characteristics of Shagawa Lake, Minnesota

Year	Natural inflow			Wastewater inflow			Combined inflow			Outflow		
	Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}	Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}	Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}	Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}
1969	83.7	15.9	4.4	1.5	3600	2550	85.2	80	45	83.7	62	19.4
1970	125.1	14.1	3.0	2.1	2600	1520	127.2	58	29	120.8	49	15.1
1971	94.0	14.7	2.8	1.7	3150	1670	95.7	71	33	93.3	48	15.9
1972	61.1	17.3	1.9	1.2	4200	2490	62.3	100	51	62.1	51	10.9
1973	102.0	16.0	2.0	1.7	310	140	103.7	20.6	4.3	91.5	47	10.3
1974	82.3	16.6	2.6	1.6	80	18.8	83.9	17.8	2.9	80.4	26	5.2
1975	65.5	14.7	1.6	1.3	44	10.9	66.8	15.3	1.8	63.5	26	3.9
1976	41.5	20.5	2.1	1.4	29	3.4	42.8	20.7	2.2	37.6	22	2.4

Table 2. Average annual total and soluble reactive phosphorus concentrations in representative tributaries to Shagawa Lake

Year	Burntside River Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}	Longstorff Creek Flow $\times 10^6$ m^3 year^{-1}	TP μg l^{-1}	SRP μg l^{-1}
1972	42.2	14.9	1.3	4.8	22.8	2.0
1973	63.9	12.7	1.2	13.3	18.8	2.0
1974	55.2	11.8	1.2	6.3	19.7	1.8
1975	44.1	12.0	1.1	5.6	18.7	1.2
1976	26.0	12.9	1.1	3.5	20.9	1.2

occurred during 1976; a high of 125.1 $\times$ 10^6 m^3 occurred during 1970 (Table 1). Annual water filling times (lake volume/inflow rate) ranged from 0.42 to 1.23 years. Water washout is rapid, so Shagawa Lake should respond promptly to a reduction in external TP supply.

The TP concentrations in the influent streams were generally low (Table 2) and unrelated to flow (Fig. 2). The mean annual TP concentration in Burntside River (Table 2) was about 1.5 times that of its source water (Burntside Lake) whose average surface (1.5 m) TP concentration was 8.3 μg l^{-1} for 1974–1975. The mean annual TP concentrations in the smaller tributaries were approximately 20 μg l^{-1}, slightly higher than those Wright (1976) reported for unperturbed streams flowing into Dogfish, Meander and Lamb Lakes, about 40 km northwest of Shagawa. Omernik (1976) reported low (mean of 14 μg l^{-1}) TP concentrations in tributaries draining forested watersheds in the eastern United States based on data from 473 subdrainage areas. SRP was barely detectable in the water of tributaries entering Shagawa Lake; annual average influent concentrations were 1–2 μg l^{-1}.

Wastewater. The wastewater supply of TP and SRP to Shagawa Lake is also summarized in Table 1. During pretreatment years the annual wastewater TP input was 5000–5500 kg of which about 60% was SRP. Treatment effectively reduced this supply to about 60 kg (about 25% was SRP). Malueg *et al.* (1975) and Larsen *et al.* (1975) described the changes in wastewater supply in more detail. During 1973, the annual average TP and SRP concentrations were higher than those during the following years because the treatment plant was not fully operational until April. The 1973 annual average reflects high values occurring during the early part of the year. Further decreases in wastewater TP content through 1976 probably occurred as plant operators became more proficient at routine plant operations.

Wastewater flow to Shagawa Lake has been low (less than 2% of total inflow) and has not exhibited the large seasonal fluctuations noted for the tributaries.

Precipitation. TP concentrations in collected rainfall varied from a low of 4 μg l^{-1} (1 September 1973) to a high of 840 μg l^{-1} (30 June 1975). Four extremely

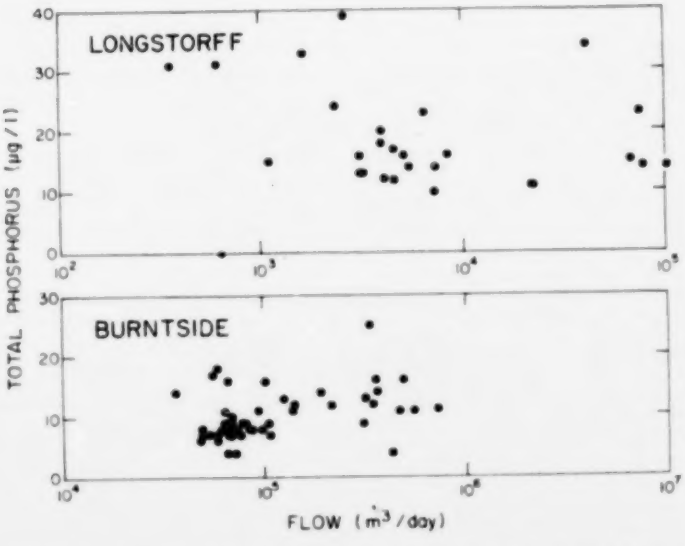

Fig. 2. Relationship between total phosphorus concentration and flow for Burntside River and Longstorff Creek.

high TP values were observed during 1975; their range was from 530–840 $\mu g\,l^{-1}$ TP. We suspect that these values are anomalous. During 1975, the collection device was located within the city of Ely; high particulate emissions (coal burning is common in the area) could have increased the TP concentration in rainfall collected to levels considerably higher than that which would fall directly on the lake. During 1973 and 1974 the precipitation collector was located 1.5 km west of the lake in an area likely to be free from the city's direct influence. Lower, perhaps more representative, TP concentrations were observed here, although some high values were observed during 1974.

The high values measured (all years) apparently relate to the time interval between collection of rainfall samples. For example, the average TP concentration (volume weighted) for samples collected at intervals >10 days was 115 $\mu g\,l^{-1}$ whereas that for intervals <5 days was 16 $\mu g\,l^{-1}$. The difference probably reflects the accumulation of dryfall in the collectors over the longer sampling intervals. The lower values (~16 $\mu g\,l^{-1}$) are more typical TP concentrations for rainfall in this area (Wright, 1976) and are similar to TP concentrations in the watershed tributaries.

Two estimates of the precipitation supply of TP to the lake are summarized in Table 3. The low estimates reflect a judgement of the acceptability of the very high TP concentrations measured in some of the samples. These (excluding the high concentrations) suggest that precipitation accounts for 10–15% of the present TP supply to the lake, although the precipitation contribution might be as high as 40% if all samples were included. The low precipitation estimates were used for projections of the lake response to TP influent reduction.

Summary of inflow characteristics. The average annual "natural" influent TP and SRP concentrations (summing all sources except wastewater) remained remarkably constant throughout the study (TP: mean = 16.2 $\mu g\,l^{-1}$, range = 14.1–20.5, SRP: mean = 2.6 $\mu g\,l^{-1}$, range = 1.6–4.4; Table 1). The TP supply from natural sources (kg $year^{-1}$) varied approximately twofold, and most of this variation can be attributed to the variation in annual water inflow.

During pretreatment years, average annual TP inflow concentrations (combining natural and wastewater) were high, ranging from about 60 to 100 $\mu g\,l^{-1}$; the variation is mainly attributed to variation in water inflow rates (Table 1). TP removal reduced these concentrations by about 80% to 18 $\mu g\,l^{-1}$ (mean of the 1974–1976 influent values, Table 1). Pretreatment influent average SRP concentrations ranged from 29 to 51 $\mu g\,l^{-1}$; treatment reduced these values to 2.3 $\mu g\,l^{-1}$, a reduction of about 93%.

Outflow characteristics

The water outflow via Shagawa River reflected the inflow seasonality; annual outflow was consistently less than the summed inflows (Table 1). The loss of water via evaporation ranged between 4.2 and 5.1 × 10^6 m^3 $year^{-1}$ (mean = 4.6 × 10^6 m^3 $year^{-1}$). The average of the yearly ratios of water loss (Shagawa River outflow plus evaporation) to summed inflows was 1.01 with a range 0.927–1.075. Groundwater inflow or outflow appears to be negligible (Malueg *et al.*, 1975), although Bradbury (1978) suggests that groundwater inflow might play an important role in controlling the chemistry of Shagawa Lake.

The measured outflow TP concentrations were not related to flow but reflected the seasonality of TP concentrations within the lake. Average annual pretreatment TP ranged from 48 to 62 $\mu g\,l^{-1}$ but was remarkably constant during 1970–1972; SRP ranged from 11 to 19 $\mu g\,l^{-1}$ (Table 1). After initiation of treatment, outflow concentrations declined to 22 $\mu g\,l^{-1}$ (TP) and 2.4 $\mu g\,l^{-1}$ (SRP) by 1976.

Estimation of phosphorus retention capacity

During 1971 and 1972, an average of 33% of the incoming TP to Shagawa Lake sedimented to the lake bottom (Table 4). Values for earlier years could not be determined because year-end data were not available for calculating changes in lake content. The ratio of outflowing TP to inflowing TP (Dillon & Rigler, 1974a) provides an estimate of R_p only if lake content does not change.

During the post-treatment years, outflowing TP has exceeded the sum of inflowing TP and the change in yearly TP content in the lake (measured by difference between the amount of TP present at the end

Table 3. Supply of total phosphorus to Shagawa Lake from precipitation falling directly on the lake

Year	Average* concentration $\mu g\,l^{-1}$	Supply kg $year^{-1}$	% of natural supply	Average† concentration $\mu g\,l^{-1}$	Supply kg $year^{-1}$	% of natural supply
1973	23	170	10	23	170	10
1974	26	190	14	53	390	25
1975	28	140	15	107	530	39
1976	32	110	13	88	300	29

* Average volume weighted concentrations excluding all samples whose TP concentration was >100 $\mu g\,l^{-1}$. During 1973, no samples rejected, during 1974, 2, during 1975, 5 and during 1976, 3 were rejected.

† Average volume weighted concentrations including all samples.

Table 4. Estimation of net annual total phosphorus deposition to, or release from, sediments in Shagawa Lake 1971–1976. Letters in parentheses indicate net annual deposition (d) or release (r)

Year	Total external supply (kg)	Total loss via outlet (kg)	Year end lake change* (kg)	Deposition (d) or release (r) (kg)	Retention coefficient
1971	6840	4680	+300	1860 (d)	0.27
1972	6240	3140	+620	2480 (d)	0.39
1973	2140	4310	−1250	920 (r)	
1974	1490	2180	+50	740 (r)	
1975	1020	1670	−280	370 (r)	
1976	890	930	+110	150 (r)	

* The yearly lake change in total phosphorus is the difference between the amount of phosphorus present at the end and at the beginning of any particular year.

and at the beginning of any year). If a net annual TP deposition had continued subsequent to treatment then outflow TP would be less than this sum; if net deposition were 0, then outflow TP would equal this sum. The positive differences suggest that the sediments are now a net internal source of TP on an annual basis (Table 4). The net annual release of TP correlates well with the flow of water through Shagawa Lake (Fig. 3) implying that for an average water year of about 83.4×10^6 m^3 year^{-1} (Table 1), a net annual sediment release of 700 kg could be expected.

A direct relationship between flow and net annual release might be expected if the sediments now regulate lake TP at concentrations higher than the average inflow concentration. Higher water flow would export greater amounts of TP from the lake; the sediments would then have to release greater amounts of TP to maintain the equilibrium value. That lake TP concentrations are generally lower than average inflow TP concentrations supports this hypothesis.

Lake changes

Table 5 summarizes Shagawa Lake's average TP (entire lake), SRP (entire lake) and chlorophyll *a* (upper 5 m) concentrations over several intervals and shows ratios expressing the concentrations over each interval as a fraction of the 1971–1972 mean for that interval. SRP displays greatest change, with a decline to about 12% of pretreatment levels during the summer interval of 1976 and to 26% of pretreatment levels during ice-cover. TP declined to about 60% of former levels during both intervals. Chlorophyll *a* (ice-free) declined to about 50% of its pretreatment concentration during 1974, but higher concentrations were measured during 1975 and 1976 although average TP levels remained nearly the same for all three years. During 1974, a greater reservoir of SRP existed than during 1975 and 1976 during the ice-free season implying that more algal biomass could have developed in 1974. Table 6 shows significant post-treatment reduction in chlorophyll *a* during May–June (50–60%) but no change during July–August.

Particularly evident are the differences between lake TP concentrations (Table 5) and those projected by equation 2 based on present influent water quality. With an average inflow TP concentration of 18 μg l^{-1} (Table 1) and $R_p = 0.33$ (Table 4), equation 2 predicts an average lake concentration of 12 μg l^{-1}. The validity of the lake TP concentration predicted by equation 2 is supported by comparison with TP concentrations in nearby lakes minimally disturbed by human activities. The average TP concentrations in Meander and Dogfish Lakes were 12 and 9 μg l^{-1} (Bradbury *et al.*, 1975). The average annual TP concentration in Burntside Lake has been about 10 μg l^{-1}.

Sufficient time has elapsed for the lake to have reached the TP levels which equation 2 projects. The time required for a lake to reach an equilibrium in response to a step change in influent concentration can be estimated from the lake's TP residence time (Vollenweider, 1975; Sonzogni *et al.*, 1976). The TP residence time can be obtained from the retention coefficient as: $\tau_p = V/(1 - R_p)Q$. For Shagawa Lake, an average value (1971 and 1972) is 0.41 year; the time required to reach 95% of an equilibrium level is $3\tau_p$, hence an average value approaching 12 μg l^{-1} would be expected within about 1.5 years after treatment, or by the end of 1974.

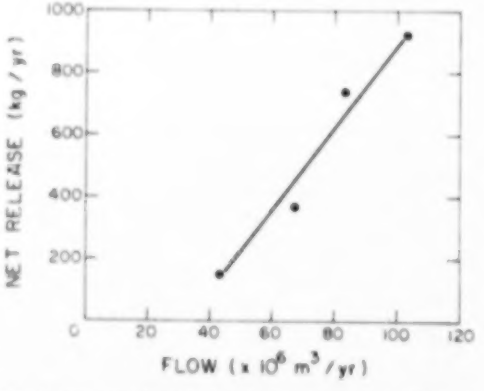

Fig. 3. Net annual total phosphorus release from Shagawa Lake sediments, 1973–1976.

Table 5. Summary of changes in TP, SRP, and chlorophyll *a* in Shagawa Lake, 1971–1976 ($\mu g\,l^{-1}$). Chlorophyll *a* values are averages over the top 5 m; TP and SRP are for entire lake. Numbers in italics are the ratios of the average concentration of any one year to the mean of the 1971 and 1972 average concentrations

Year	Annual average TP	Annual average SRP	Ice-covered interval TP	Ice-covered interval SRP	Ice-free interval TP	Ice-free interval SRP	Chlorophyll *a*
1971	47.4	21.2	36.5	24.5	54.5	19.2	20.0
	0.93	*0.99*	*0.92*	*0.91*	*0.94*	*1.12*	*0.90*
1972	54.1	21.7	42.7	29.4	60.9	15.0	24.3
	1.07	*1.01*	*1.08*	*1.09*	*1.06*	*0.88*	*1.10*
1973	48.7	16.2	51.4	32.5	50.8	10.4	21.2
	0.96	*0.76*	*1.30*	*1.21*	*0.88*	*0.61*	*0.96*
1974	31.4	9.3	24.6	12.8	35.7	6.9	11.3
	0.62	*0.43*	*0.62*	*0.47*	*0.62*	*0.40*	*0.51*
1975	29.3	6.1	23.7	13.2	34.6	2.5	16.3
	0.58	*0.28*	*0.60*	*0.49*	*0.60*	*0.15*	*0.74*
1976	29.5	4.5	19.4	7.1	35.7	2.1	18.5
	0.58	*0.21*	*0.49*	*0.26*	*0.62*	*0.12*	*0.84*

The estimated equilibrium TP content can be used to predict chlorophyll *a* concentrations in Shagawa Lake. Dillon & Rigler (1974b) related average trophogenic summer (mid-May to early September) chlorophyll *a* to springtime TP concentration as: $\log_{10}$ (chlorophyll *a*) = 1.449 log [P meas] − 1.136. Using the projected TP equilibrium value of $12\,\mu g\,l^{-1}$ in Shagawa Lake, average summer chlorophyll *a* concentrations of about $2.7\,\mu g\,l^{-1}$ are predicted. In Burntside Lake where springtime TP concentrations were 10 (1974) and 9.2 (1975) $\mu g\,l^{-1}$, summer chlorophyll *a* concentrations were 3.2 (1974) and 2.9 (1975) $\mu g\,l^{-1}$. Dillon & Rigler's relationship predicts 2.06 and $1.82\,\mu g\,l^{-1}$ respectively, for Burntside Lake; the observed values fall within the rather wide 95% confidence limits Dillon & Rigler calculated for the relationship. The projected $2.7\,\mu g\,l^{-1}$ chlorophyll *a* concentrations in Shagawa Lake appear valid for the projected TP content. The considerably higher chlorophyll *a* concentrations correspond to the higher than projected TP concentrations.

Changes in the seasonal pattern of average lake TP in Shagawa are summarized in Figs 4 and 5. The pretreatment TP pattern is characterized by slowly increasing concentrations under ice-cover developing from the supply of wastewater TP. Prior to ice-out, the lake content increased rapidly over a short interval, probably in response to the development of anaerobic conditions in the bottom water and associated release of sediment TP. Shortly after ice-out, TP declined from these spring peaks to yearly low concentrations during June, then increased to yearly high values by August and early September prior to initiation of fall circulation. A steady decline followed throughout the autumn until early winter concentrations were attained. Following treatment, the general pattern has remained with some differences. TP concentrations measured during ice-cover remained stable rather than increased, and the spring pulse has

Table 6. Changes in chlorophyll *a* (top 5 m) in Shagawa Lake during two intervals: May–June and July–August. Numbers in italics are the ratios of the average concentration of any one year to the mean of the 1971 and 1972 average concentrations (after Smith *et al.*, in prep)

Year	May–June		July–August	
1971	17.0	*1.10*	23.0	*0.82*
1972	14.1	*0.91*	32.9	*1.18*
1973	16.1	*1.04*	24.3	*0.87*
1974	11.4	*0.73*	15.5	*0.55*
1975	7.8	*0.50*	23.3	*0.84*
1976	6.4	*0.41*	33.4	*1.20*

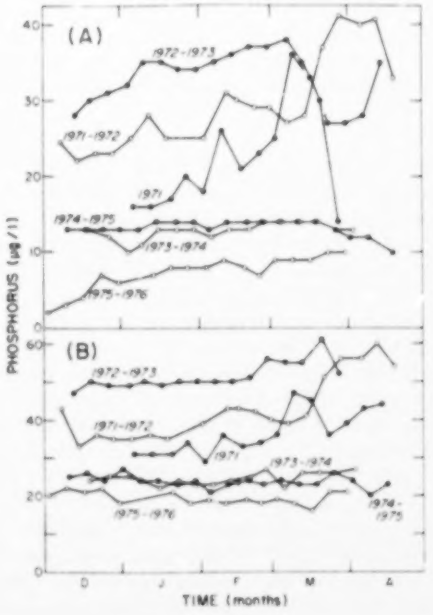

Fig. 4. Average (volume weighted) total and soluble reactive phosphorus concentrations in Shagawa Lake during periods when the lake was ice-covered. Panel A is SRP; panel B is TP.

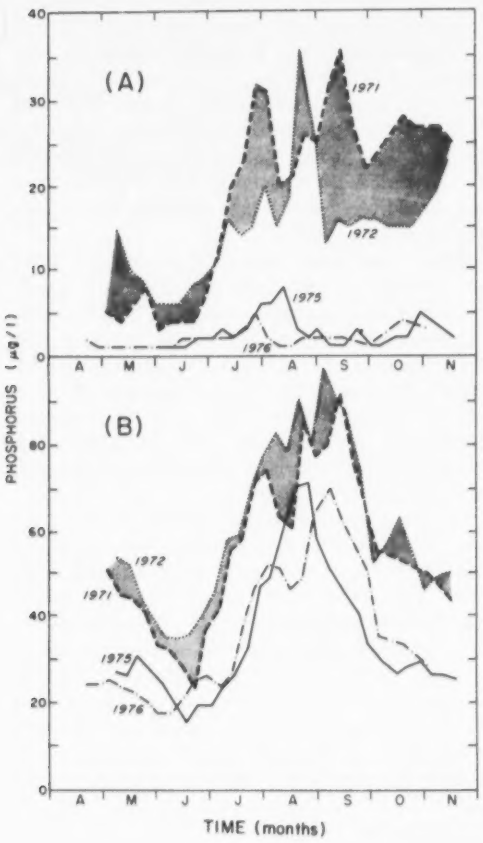

Fig. 5. Average (volume weighted) total and soluble reactive phosphorus concentrations in Shagawa Lake during periods when the lake was ice-free. Panel A is SRP; panel B is TP. Shaded areas emphasize pretreatment concentrations.

been significantly reduced. The summer pulse persists. Concentrations throughout the year have been consistently and significantly lower than those measured before treatment, as shown in Figs 4 and 5.

The seasonal pretreatment SRP pattern was similar to that of TP with increasing concentrations during the ice-covered intervals, a spring pulse followed by a dramatic reduction, then a summer increase (Figs 4 and 5). SRP concentrations generally remained high and variable through the summer with rapid, severe fluctuations seen during the late summer. During the fall no consistent decline occurred, as was observed for TP.

Subsequent to treatment significant changes occurred in the seasonal SRP pattern (Figs 4 and 5). Concentrations during the ice-covered interval stabilized at about $10\,\mu g\,l^{-1}$, increasing from low values during the fall. The spring pulse has been minimized, and nearly complete depletion has occurred during early summer. The characteristic summer pulse has almost disappeared.

The pretreatment seasonal chlorophyll *a* cycle generally followed a pattern of low values during the ice-covered interval increasing to spring peaks during or just after ice-out (Fig. 6). In 1973, chlorophyll *a* attained a spring peak prior to ice-out because the lake was covered with clear ice (U.S. EPA, unpublished). Chlorophyll *a* typically declined from spring peaks to early summer minima consistent with the TP minima then increased to yearly maxima during July and August. A steady decline occurred during the autumn and early winter months, with minimum concentrations of $<1\,\mu g\,l^{-1}$ seen by late winter. This pattern has remained basically unaltered in the years subsequent to treatment, except during spring and early summer when concentrations have been significantly lower than during pretreatment years (Table 6). The summer bloom (1976) has attained concen-

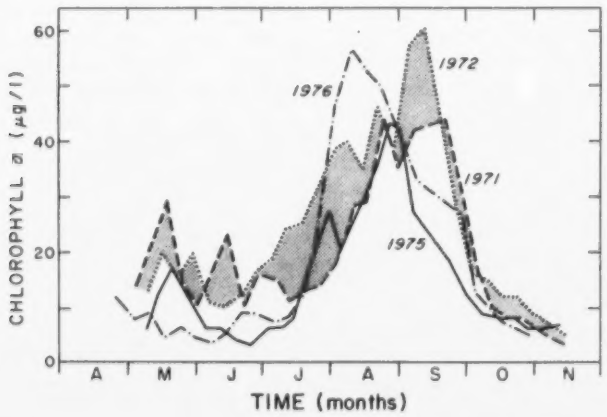

Fig. 6. Average (volume weighted) chlorophyll *a* concentrations in upper 5 m in Shagawa Lake during the ice-free period. Shaded areas emphasize pretreatment concentrations.

Table 7. Parameter estimates for equation (3)

Ice-covered season	Ice-free season
$\sigma_p^w = 0.035$ week^{-1}	$\sigma_p^s = 0.072$ week^{-1}
$C_1 = 280$ kg week^{-1}	$C_2 = 500$ kg week^{-1}
$t_1 =$ week 12	$t_3 =$ week 25
$t_2 =$ week 14	$t_4 =$ week 34

trations similar to those of pretreatment years but the duration of the bloom has been shorter.

Modified seasonal model

The parameter estimates for the modified mass balance model (equation 3) for the ice-free and ice-covered intervals are summarized in Table 7. A comparison of model output using the parameter estimates in Table 7 with average lake TP concentrations during 1971 (corroboration interval) and 1972 (estimation interval) shows a reasonably good fit (Fig. 7). There is about a 25% reduction in the model error over the year-long corroboration interval as compared with the year-long estimation interval; the sum of the squared deviations for the calibration interval was 2643 while that for the corroboration interval was 1935. The mean deviations for both the estimation and corroboration intervals were low, 1.2 and $-1.2\ \mu g\ l^{-1}$ respectively (Table 8).

The model responds rapidly to changes in input functions. If the same water and nutrient inputs are repeated year after year in the model, a steady state is achieved within two years, i.e. successive years have identical patterns in weekly lake TP concentrations. This is consistent with the prompt response predicted by equation 1 to changes in TP supplies.

Model projections suggest that an immediate response should have occurred in the lake when tertiary wastewater treatment began in early 1973 (Fig. 8). Deviations from measurements are evident, particularly during early 1973 when the TP concentrations generated by the model decrease rapidly but lake TP increased slightly. Since water flow through the lake is low during the ice-covered periods, this decrease in modeled TP occurs via deposition. With $\sigma_p^w = 0.00$, considerably closer agreement between model and data occurs during this period (Fig. 8). Estimates of σ_p^w used in the model could be high during the period when the model was fit. The model assumes that all TP entering the lake is mixed throughout the lake instantly. If a fraction of the wastewater TP supply were not mixed instantly, but instead deposited rapidly, a truer picture of the wastewater supply to the lake might include a lower value than that used in the model. A lower deposition coefficient (perhaps zero) would be necessary to compensate for the lowered wastewater supply.

Model runs which assume treatment and either $\sigma_p^w = 0.035$ or $\sigma_p^w = 0.00$ (with $\sigma_p^s = 0.072$) also consistently underestimate the data during the period from about August 1973 (week 33), to June 1974 (week 25). These differences can be traced to a difference in timing of the termination of the summer pulse. In 1973, the measurements suggest that the pulse continued for several weeks longer than assumed in the model. The effects of this timing discrepancy are propagated through June, 1974 (week 25).

The model, either with $\sigma_p^w = 0.035$ or $\sigma_p^w = 0.00$ and $\sigma_p^s = 0.072$, tracks the data from mid-1974 through 1976 reasonably well. However, during early 1975, with $\sigma_p^w = 0.035$, consistent underestimates occur, and during early 1976, with $\sigma_p^w = 0.00$, consistent overestimates are seen. Closer agreement would have been obtained with $\sigma_p^w = 0.00$ during early 1976 if the model had not switched to $\sigma_p^w = 0.00$ on week 46, 1975, but had delayed 4 weeks (Fig. 8). The TP measurements during the early part of each year after treatment (weeks 1–12) indicate that a minimum amount of deposition occurs, consistent with the model using $\sigma_p^w = 0.00$.

Projections of the TP patterns which could have occurred if treatment had not been instituted reflect the stable TP pattern which had been established prior to treatment (Figs 7 and 8). These projections, when compared with the data, clearly show the effect of treatment in reducing TP concentrations in the lake after 1973 (Fig. 8). Interestingly, the TP concentrations in the lake during the first year after treatment began are fit quite well by the model which assumes no treatment. This suggests that the lake did not begin to respond to treatment until fall 1973.

The model projections are also summarized in

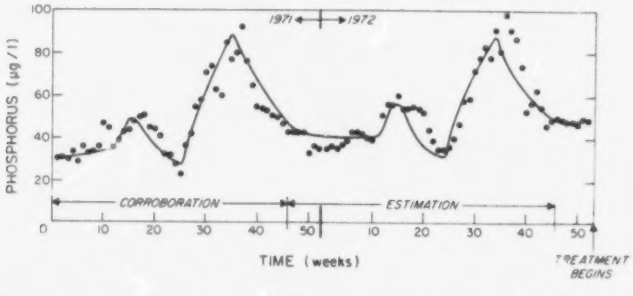

Fig. 7. Comparison of output from equation 3 (parameter estimates listed in Table 7) with total phosphorus concentrations (volume weighted average) in Shagawa Lake. Equation 3 was fit to the 1972 data; 1971 data were used to corroborate the model.

Table 8. Mean annual and seasonal TP concentrations ($\mu g\,l^{-1}$) calculated from model projections (equation 3); (a) no treatment, (b) treatment, $\sigma_p^w = 0.035\,week^{-1}$, (c) treatment $\sigma_p^w = 0.00\,week^{-1}$. Mean deviation = Σ(data − model) number points

Year	Annual average TP	Annual average Mean deviation	Ice-cover TP	Ice-cover Mean deviation	Ice-free TP	Ice-free Mean deviation
(a)						
1971	48.2	−1.2			56.5	−1.6
1972	52.1	1.2	44.4	−1.7	58.1	2.7
1973	49.9	−1.2	52.0	0.3	51.6	−1.5
1974	49.8	−18.3	39.0	−12.9	55.8	−19.8
1975	54.2	−25.0	47.1	−23.0	58.8	−24.2
1976	58.3	−28.2	49.9	−30.3	62.7	−27.0
(b)						
1973	36.7	12.4	44.7	7.3	38.1	12.9
1974	27.9	3.3	13.8	12.4	37.7	−1.5
1975	30.0	−1.0	19.3	5.0	38.3	−3.7
1976	32.8	−2.8	20.8	−0.8	40.3	−4.6
(c)						
1973	43.3	5.4	53.8	−1.5	41.8	8.4
1974	31.3	0.2	19.0	7.2	39.3	−3.3
1975	34.8	−5.6	26.6	−2.5	40.7	−6.1
1976	38.8	−8.7	28.2	−8.6	43.8	−8.7

Table 8 as average annual and seasonal TP concentrations with measures of model deviations from lake values. In the absence of treatment, the annual average TP concentration predicted by the model for 1973–1976 seems fairly stable at $\sim 50\,\mu g\,l^{-1}$, and slightly higher during the ice-free season. The highest mean annual concentration which the model produced occurred in 1976. This high concentration reflects the extremely low flow for that year (Table 1). Model runs assuming treatment suggest that the annual averages might be expected to decrease to $\sim 30\,\mu g\,l^{-1}$ (if $\sigma_p^w = 0.00$). Lower values, on the order of $20\,\mu g\,l^{-1}$ (somewhat higher if $\sigma_p^w = 0.00$), might be expected during the ice-covered season, and an average of about $40\,\mu g\,l^{-1}$ would be seen during the ice-free season. These values are generally consistent with those summarizing the lake's response (Table 5).

Mean deviations for model runs in the absence of treatment display the progressive improvement in the lake both during ice-free and ice-covered periods, suggesting that the lake P concentrations have declined about $30\,\mu g\,l^{-1}$ from what they might have been in

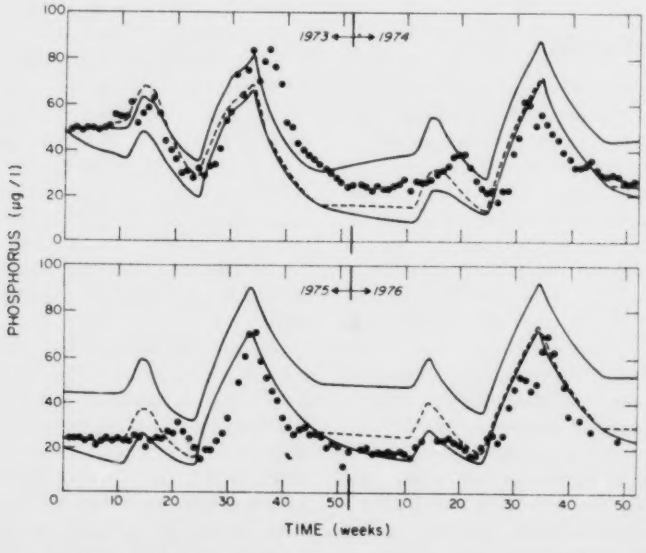

Fig. 8. Comparison of output from equation 3 with total phosphorus (volume weighted average) data for the years subsequent to treatment. Upper solid line is projection as if treatment had not been instituted; lower solid line is projection incorporating external loading reduction; dashed line is the same run except $\sigma_p^w = 0.00$. Coefficient values used are summarized in Table 7.

Table 9. Dates indicate onset of oxygen depletion ($<1.0\,mg\,l^{-1}$) each summer and reappearance of oxygenated conditions ($>1.0\,mg\,l^{-1}$) after fall circulation (1971–1976) in Shagawa Lake

Year	Onset of O_2 depletion	Reoxygenation
1971*	22 June	28 Sept
1972	13 June	19 Sept
1973	19 June	11 Sept
1974*	18 June	27 Aug
1975	17 June	9 Sept
1976	2 June	8 Sept

* With the exception of 27 July 1971 and 30 July 1974, when strong windstorms aerated the entire lake the week prior to these sampling dates.

the absence of treatment (Table 8). The mean deviations for the 1973 annual average as well as for the seasonal averages show the lack of immediate response observed in Fig. 7.

The goodness-of-fit of the model through 1976, assuming treatment, does not allow us to reject some important assumptions embodied in the model. One assumption is that release of TP from the sediments is time-invariant with respect to release rate and the time interval of occurrence. It appears that treatment has not yet influenced the pattern of TP release and that the characteristics which the lake now exhibits are mainly controlled by this release. The second assumption is that the TP deposition rate σ_p, is not coupled to the TP level in the lake, but remains constant. The model projections imply that this assumption is adequate for deposition during the summer (σ_p^s), but wintertime deposition rates might drop to near zero in posttreatment years.

DISCUSSION

A significant and prompt reduction in the TP and SRP concentrations in Shagawa Lake has been measured in the years after influent P curtailment, but the TP concentration has not declined to levels projected from present influent water quality (equation 2). The major deviation has occurred during the months of July and August when an internal discharge of P has increased the lake TP content by about 2500 kg, increasing the average lake TP concentration $\sim 50\,\mu g\,l^{-1}$. Just prior to this discharge, average lake TP concentrations have been $<20\,\mu g\,l^{-1}$, slightly higher than equation 2 projects. We provided evidence in another paper (Larsen *et al.*, 1979) that a significant fraction of the net lake increase could be attributed to release from the profundal zone sediments during periods when these sediments are overlain by anaerobic or nearly anaerobic water. The sequence typically occurring in Shagawa Lake is: (1) Anaerobic conditions develop in the bottom water in mid-June (Table 9) and remain until fall circulation unless significant wind activity aerates the entire lake. (2) As anaerobic conditions develop, TP concentrations in the bottom water increase corresponding to release rates on the order of 7–10 mg $P\,m^{-2}\,day^{-1}$. The iron content of the sediments is high (5% dry wt) and increases in TP in the deeper water are directly correlated with increases in iron, implying that TP bound to iron oxides is released when these oxides dissolve as the redox potential declines (Mortimer, 1941, 1942). (3) TP in the deeper water is transported upward by two mechanisms: (a) turbulent diffusion, and (b) entrainment. Typically, by June, a thermocline has developed in Shagawa Lake at about 5 m, but downward heat transport is not stopped. The heat content of the lower water can increase significantly during July corresponding to approximate Fickian diffusion coefficients of 0.01–0.02 $cm^2\,s^{-1}$ (calculated by equation 8 in Sweers, 1970). Steep phosphorus gradients develop below the thermocline; the average gradient for July, 1974 and 1975 was $\sim 70\,\mu g$ $TP\,l^{-1}\,m^{-1}$; thus 6–12 mg $m^{-2}\,day^{-1}$ might be transported vertically per week by turbulent diffusion. During late July through August, the depth of the thermocline typically deepens, sometimes dramatically in a few days if winds are high, entraining phosphorus. During 1974, between 26 and 29 July, the mixed zone deepened from 5–9 m in response to winds of 15–20 $km\,h^{-1}$. An increase in TP of 1250 kg (corresponding to an increase of about $30\,\mu g\,l^{-1}$) occurred in the top 5 m within one week. In general, extension of the mixed zone is less dramatic, but its occurrence is consistent each year (Larsen *et al.*, 1979).

Shagawa Lake's basin is characterized by steep sides to about 6.5 m and a relatively flat profundal zone with several depressions. The area of the profundal zone is about 55% of the total lake area of 9.5 km^2. Thus near bottom water high in TP can occur close to the surface water facilitating transport of TP by both above mechanisms.

Larsen *et al.* (1979) also describe other mechanisms which could contribute TP during the summer months, but ascribe the major input to the profundal zone sediments. The magnitude of the pulse has shown no significant decrease in the years since treatment. From 1971–1975, TP in the lake increased an average of 2500 kg (Larsen *et al.*, 1979); during 1976 the increase was ~ 2400 kg. The fit of equation 3 to post-treatment data is in accord with these observations.

The algal community has responded to the increased TP content in the lake during July and August with a significant increase in biomass. Since a sizable reservoir of SRP existed during the summers of pretreatment years (surface concentrations were generally $>20\,\mu g\,l^{-1}$ during July and August), algal growth was not controlled by SRP availability at that time; significant increases in biomass could have occurred using remaining SRP. Subsequent to treatment, this SRP reservoir was substantially depleted; the size of the algal crop is now probably controlled by the availability of SRP, but enough still becomes

available during the summer to support algal biomass as large as that which occurred during the pretreatment periods. Further reductions in lake TP could be expected to produce reductions in algal biomass.

One of the main advantages of using equations 1 and 2 is that the average TP content of a lake can be predicted with relatively little data: lake morphometry, water and TP inflows and outflows. However, the characteristics which Shagawa Lake exhibited prior to treatment as well as its response to TP reduction demonstrate that internal mechanisms can exert a significant influence over how a lake responds to influent TP reduction. Unfortunately it is not possible to predict with any degree of confidence the nature or extent of control by internal mechanisms. Hence for lakes where this control is important these properties must be determined before successful projections can be obtained.

Numerous cases have been documented in which significant external phosphorus supply reduction has caused significant change in recipient lake waters as might be projected from equations 1 and 2: Lake Washington (Edmondson, 1972, 1977), Lakes Kegonsa and Waubesa (Sonzogni *et al.*, 1976); Onondaga (Murphy, 1973), Little Otter Lake (Michalski & Conroy, 1973), Gravenhurst Bay (Michalski *et al.*, 1975; Dillon *et al.*, 1978); Lake 304, Experimental Lakes Area (Schindler & Fee, 1974); Lower St. Regis (Fuhs *et al.*, 1977); and Lyngby Sø, Bagsvaerd Sø, and Pedersburg Sø (Mathiesen, 1971).

However, several cases besides Shagawa occur in which significant control over lake response is apparently exerted by internal mechanisms. Lake Sammamish has not responded detectably to a 30% reduction in external TP supply (Emery *et al.*, 1973; Welch, 1977). Several Swedish lakes had not responded positively or had actually deteriorated after significant curtailment of TP supplies (through diversion of wastewater TP or treatment for wastewater TP removal) although sufficient time had elapsed for response to have occurred (Ryding & Forsberg, 1976). Lorenzen (1973) described how continued deterioration can occur even after TP supplies have been reduced. Although lake TP concentrations have declined from about 470 $\mu g\,l^{-1}$ to 150 $\mu g\,l^{-1}$ in Lake Norrviken as a result of wastewater diversion, the response of the lake has been regulated to a significant extent by TP release from sediments (Ahlgren, 1976). Lake Trummen did not recover after wastewater inflow curtailment in 1957–1959 but required dredging during 1970–1971 to remove nutrient rich sediment (Björk, 1973; Andersson *et al.*, 1973).

In addition, internal phosphorus supplies have been implicated in controlling the productivity of several lakes: Klamath Lake, Oregon (Wildung *et al.*, 1977); Twin Lakes, Ohio (Cooke *et al.*, 1977); Lake Erie (Burns & Ross, 1972). For these lakes, a response to TP supply reduction would probably be regulated, in part, by internal mechanisms.

The delayed response of Sammamish was attributed to a regulation of TP levels in the lake, in part, by P–iron interactions associated with seasonal changes from aerobic to anaerobic conditions in the deeper waters (Welch, 1977). Ryding & Forsberg (1976) cite significant feedback of phosphorus from the sediments of some of the shallower Swedish lakes in controlling recovery but suggest that this feedback is not so important in controlling the recovery of larger stratified lakes. Significant feedback of TP from the sediments of Lake Norrviken has inhibited its recovery although the TP flux from the sediments has been decreasing since treatment began (Ahlgren, 1976).

It is apparent that even when external TP supplies to lakes have been curtailed to a significant extent, corresponding decreases in lake TP levels, and hence lake productivity, might not occur within an acceptable time scale. Thus from a water quality management point of view, it is important to analyze the potential response of a lake to influent TP reduction before a huge investment is made in reducing nutrient supplies. To achieve a desired water quality within a reasonable length of time it might also be necessary to institute in-lake remedial measures.

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